LCA Methodology

Impact Categories for Life Cycle Assessment Research of Seafood Production Systems: Review and Prospectus

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DOI: http://dx.doi.org/10.1065/lca2006.09.275

Please cite this paper as: Pelletier NL, Ayer NW, Tyedmers PH, Kruse SA, Flysjo A, Robillard G, Ziegler F, Scholz AJ, Sonesson U (2007): Impact Categories for Life Cycle Assessment Research of Seafood Production Systems: Review and Prospectus. Int J LCA 12 (6) 414–421

Abstract

Goal, Scope and Background. In face of continued declines in global fisheries landings and concurrent rapid aquaculture development, the sustainability of seafood production is of increasing concern. Life Cycle Assessment (LCA) offers a convenient means of quantifying the impacts associated with many of the energetic and material inputs and outputs in these industries. However, the relevant but limited suite of impact categories currently used in most LCA research fails to capture a number of important environmental and social burdens unique to fisheries and aquaculture. This article reviews the impact categories used in published LCA research of seafood production to date, reports on a number of methodological innovations, and discusses the challenges to and opportunities for further impact category developments.

Main Features. The range of environmental and socio-economic impacts associated with fisheries and aquaculture production are introduced, and both the commonly used and innovative impact categories employed in published LCA research of seafood production are discussed. Methodological innovations reported in agricultural LCAs are also reviewed for possible applications to seafood LCA research. Challenges and options for including additional environmental and socioeconomic impact categories are explored.

Results. A review of published LCA research in fisheries and aquaculture indicates the frequent use of traditional environmental impact categories as well as a number of interesting departures from the standard suite of categories employed in LCA studies in other sectors. Notable examples include the modeling of benthic impacts, by-catch, emissions from anti-fouling paints, and the use of Net Primary Productivity appropriation to characterize biotic resource use. Socio-economic impacts have not been quantified, nor does a generally accepted methodology for their consideration exist. However, a number of potential frameworks for the integration of such impacts into LCA have been proposed.

Discussion. LCA analyses of fisheries and aquaculture call attention to an important range of environmental interactions that are usually not considered in discussions of sustainability in the seafood sector. These include energy use, biotic resource use, and the toxicity of anti-fouling paints. However, certain important impacts are also currently overlooked in such research. While prospects clearly exist for improving and expanding on recent additions to environmental impact categories, the nature of the LCA framework may preclude treatment of some of these impacts. Socio-economic impact categories have only been described in a qualitative manner. Despite a number of challenges, significant opportunities exist to quantify several important socio-economic impacts.

Conclusion. The limited but increasing volume of LCA research of industrial fisheries and aquaculture indicates a growing interest in the use of LCA methodology to understand and improve the sustainability performance of seafood production systems. Recent impact category innovations, and the potential for further impact category developments that account for several of the unique interactions characteristic of fisheries and aquaculture will significantly improve the usefulness of LCA in this context, although quantitative analysis of certain types of impacts may remain beyond the scope of the LCA framework. The desirability of incorporating socio-economic impacts is clear, but such integration will require considerable methodological development.

Recommendations and Perspectives. While the quantity of published LCA research for seafood production systems is clearly increasing, the influence this research will have on the ground remains to be seen. In part, this will depend on the ability of LCA researchers to advance methodological innovations that enable consideration of a broader range of impacts specific to seafood production. It will also depend on the ability of researchers to communicate with a broader audience than the currently narrow LCA community.

Keywords: Aquaculture; fisheries; impact categories; LCA; LCIA; seafood

Introduction

According to the United Nations Food and Agriculture Organization (FAO) the global production of seafood from fisheries and aquaculture reached 133 million tonnes, and provided direct employment to an estimated 38 million people, in 2002 (FAO 2004). With many traditional fisheries depleted, over-exploited, or fully exploited, it appears that global carrying capacity for seafood production has been reached or even exceeded (Pauly et al. 2002, Myers and Worm 2003, Worm and Myers 2004), and that restrictive management regimes are imperative (Pauly et al. 2003). In response to this decline and increasing demand for seafood products, aquaculture has become the fastest growing animal food-producing sector (FAO 2004). At present, approximately 40% of all consumed fish and shellfish are farmed, and a 70% increase in aquaculture production by the year 2030 is predicted (FAO 2004).

Both fisheries and aquaculture have been implicated in a variety of environmental and socio-economic impacts. Besides direct impacts to targeted stocks (Pauly et al. 2002, Christensen et al. 2003, Myers and Worm 2003), some fish-

eries have been criticized for generating substantial by-catch (Alverson et al. 1994, Glass 2000), the disturbance and displacement of benthic communities (Johnson 2002, Chuenpagdee et al. 2003) and the alteration of trophic dynamics (Jackson et al. 2001). Criticisms of aquaculture, which are largely associated with the intensive cultivation of highly-valued shrimp and salmon species, include eutrophication of local water bodies (Folke et al. 1992), deterioration of the benthos (Findlay et al. 1995, Paez-Azuna 2001), introduction of genetic material into compromised conspecific populations (Unum and Fleming 1997, Youngson and Verspoor 1998, Fleming et al. 2000), the amplification and retransmission of diseases and parasites to the wild (Arkoses et al. 2005), the discharge of pharmaceuticals and other chemicals into the marine environment (Hastein 1995), the depletion of wild stocks through broodstock or seed harvesting (Mungkung et al. 2006) , and a dependence on fish-based feeds (Naylor et al. 2000, Naylor and Burke 2005).

While discourse regarding the environmental repercussions of seafood production is often dominated by such proximate biological interactions, other research indicates that the material and energetic demands of both industrial fisheries and aquaculture can also precipitate considerable impacts. In the fisheries sector, these include the environmental and socio-economic costs of fishing vessel construction and maintenance (Watanabe and Okubo 1989, Hayman et al. 2000), the provision of fishing gear (Ziegler et al. 2003), the discharge of wastes and loss of fishing gear at sea (Derraik 2002), the use of fossil fuels in vessels (Ziegler and Hansson 2003, Thrane 2004b, Tyedmers 2004, Hospido and Tyedmers 2005, Tyedmers et al. 2005), and the transportation and processing of landings (Karlsen and Angelfoos 2000, Andersen 2002). In aquaculture, these secondary impacts are largely related to the high material and energetic demands associated with the provision of concentrated feed for intensive production systems and the maintenance of water quality in closed containment systems (Tyedmers 2000, Troell et al. 2004, Papatryphon et al. 2004). Furthermore, many of these impacts may also threaten the health, social, economic and cultural well-being of adjacent human communities as well as consumers of seafood products (Anderson and Fong 1997, Hites et al. 2004).

The rapid expansion of the aquaculture sector and the vulnerability of global fisheries to further degradation underscore the urgent need to understand and manage the environmental and social interactions of seafood production systems. Life Cycle Assessment (LCA) provides a convenient, standardized means of quantifying and describing many of these interactions and targeting specific process and product improvements. However, the efficacy of this tool in providing comprehensive insights as to the scope and scale of such impacts will ultimately be limited by the availability of appropriate impact categories.

1 Impact Assessment in LCA

The magnitude and significance of environmental or social costs associated with specific life cycle activities are identified during the Life Cycle Impact Assessment (LCIA) phase (Pennington et al. 2004). This is achieved by quantitatively expressing the results of the Life Cycle Inventory (LCI) using impact categories (classes representing environmental issues of concern) and their associated category indicators (quantifiable resources/emissions/substances representing each impact category) (Guinee et al. 2001). ISO defines both mandatory and optional elements of the LCIA framework. Mandatory elements are: the selection of impact categories, category indicators and characterization models; the assignment of LCI results (classification); and the calculation of category indicator results (characterization). Optional elements are: calculation of the magnitude of category indicator results relative to reference information (normalization); grouping; weighting; and data quality analysis (Guinee et al. 2001).

SETAC standards (Consoli et al. 1993) define four key impact categories for LCA: ecological health, human health, resource depletion, and social welfare. ISO (2003) does not include social welfare as a mandatory impact category, but does state that where existing impact categories, category indicators, and characterization models are not sufficient to satisfy the defined goal and scope of the research, new ones must be defined. The ISO standards further require that the selection of impact categories reflects a comprehensive set of environmental issues relevant to the system under study.

These impact categories are very broad and, depending on the nature of the study, are generally sub-divided to represent more specific impacts. The European Environment Agency (Jensen et al. 1999) identifies abiotic resources, biotic resources, land use issues, global warming, stratospheric ozone depletion, ecotoxicological impacts, human toxicological impacts, photochemical oxidant formation, acidification, eutrophication, and the human work environment as priority impact categories. With the exception of human work environment, most of these categories are commonly employed in published LCA studies (**Table 1**).

Table 1: Impact categories commonly employed in published LCA research

Impact Category	Description of Impacts
Global Warming	Contributes to atmospheric absorption of infrared radiation
Acidification	Contributes to acid deposition
Eutrophication	Provision of nutrients contributes to biological oxygen demand
Photochemical Oxidant Formation	Contributes to photochemical smog
Aquatic/Terrestrial Ecotoxicity	Contributes to conditions toxic to flora and fauna
Human Toxicty	Contributes to conditions toxic to humans
Energy Use	Contributes to depletion of non-renewable energy resources
Abiotic Resource Use	Contributes to depletion of non-renewable resources
Biotic Resource Use	Contributes to depletion of renewable resources
Ozone Depletion	Contributes to depletion of stratospheric ozone

1.1 Commonly employed impact categories in seafood LCAs

While Life Cycle Assessment in the agriculture sector is relatively well established (Andersson et al. 1994, Andersson 2000), the application of this tool for evaluating seafood production systems is a more recent phenomenon. To date, LCA researchers in fisheries and aquaculture have variously examined Norwegian cod fisheries (Ellingson and Aanondsen 2006), Spanish tuna fisheries (Hospido and Tyedmers 2005), Danish fish products (Thrane 2004a, 2006), Swedish cod products (Ziegler et al. 2003), Finnish trout production (Seppala et al. 2001), farmed salmon (Ellingson and Aanondsen 2006), farmed Thai shrimp products (Mungkung 2005), and French trout farms and feeds (Papatryphon 2003, 2004), The conclusions generated by these studies suggest that the LCA framework is well-suited to informing eco-efficiency measures in fisheries and aquaculture, and that the life cycle perspective holds considerable promise for informing seafood product-oriented environmental policy. However, it is apparent that the development of appropriate impact categories will be essential to arriving at more comprehensive evaluations of the environmental and social interactions of seafood production.

Of the studies reviewed (**Table 2**), all employed global warming, acidification, and eutrophication as impact categories. In some cases, eutrophication has been sub-divided to reflect the relative contributions of N and P emissions (Ziegler et al. 2003), or emissions to land or water (Seppala et al. 2001). As with photochemical oxidant formation, abiotic resource use (alternately referred to as energy use, abiotic depletion, or depletion of fossil fuels) has also been commonly, if not consistently, employed (Seppala et al. 2001, Papatryphon et al. 2003 and 2004, Ziegler et al. 2003, Mungkung 2005). Toxicity-related emissions have been variously quantified in terms of marine, aquatic, freshwater, terrestrial, and human toxicological impacts. Papatryphon et al. (2003, 2004) were alone in accounting for biotic resource consumption. The use of these categories is consistent with the impact category choices typical of LCA research in other sectors. In so far as accounting for broad-scale environmental impacts characteristic of human industrial activities, these choices are certainly defensible. However, if the goal is to arrive at more comprehensive assessments and opportunities for improvement in seafood production systems then further impact category development is desirable.

1.2 Challenges to seafood production-specific impact category development

Perhaps due to its origin and evolution in the context of traditional industrial production systems, the standard suite of impact categories used in most LCAs accounts for an important but limited range of environmentally significant chemical parameters. In contrast, the life cycle impacts of seafood production often encompass a much greater range of environmental concerns. A major challenge facing LCA researchers of seafood production systems is to develop novel impact categories appropriate to quantifying the environmental and social interactions characteristic of specific seafood production technologies and, where this is not possible, to clearly articulate the limitations of the information generated.

LCA is most often used to describe product life cycle impacts that contribute to broad-scale, global environmental concerns such as climate change, ozone depletion, and abiotic resource use. The physical/chemical pathways that contribute to these impacts are reasonably well-known and relatively easy to quantify. Standardized analyses and comparisons between production systems and technologies are therefore largely straight-forward, whatever the geographical context. For example, a tonne of greenhouse gas emissions will contribute to climate change in a specific manner, regardless of where the emissions occur. This is similarly true of the release of ozone depleting substances and the consumption of non-renewable resources. With certain other traditional impact categories, including eutrophication, acidification, and marine ecotoxicity, the relationships between local emissions and global impacts are less clear. In the case of eutrophication, for example, the actual eutrophication potential of nutrient emissions will be very much context specific, depending on a wide variety of variables which together define the sensitivity and assimilatory capacity of the ecosystem in question. Nonetheless, the pathways leading to eutrophication impacts are well-known and easy to quantify.

There are numerous other environmental concerns that manifest only at local levels but also contribute to globally-recognized issues. These include biodiversity loss, biotic resource depletion, and the erosion of ecosystem structure and function through habitat alteration. However, such impacts are often much more difficult to quantify, and accepted methodologies for use in ISO-compliant Life Cycle Assessment

Table 2: Impact categories and category indicators employed in selected LCA research of seafood production systems

Author	Research	GW	Ac	Eu	PO	AE	TE	Ht	EU	BR	OD
Hospido and Tyedmers (2005)	Spanish tuna fisheries	CO_2	SO_2	PO_4	C_2H_4	1,4 DCB		1,4 DCB			CFC
Mungkung (2004)	Thai shrimp aquaculture	CO_2	SO_2	PO_4	C_2H_4	1,4 DCB	1,4 DCB	1,4 DCB	Sb		CFC
Papatryphon et al. (2003, 2004)	French trout aquaculture and feeds	CO_2	SO_2	PO_4					MJ	NPP	
Thrane 2004a, 2006	Danish fisheries	CO_2	SO_2	PO_4	C_2H_4	H_2O			MJ		CFC
Ziegler et al. (2003)	Swedish cod fisheries	CO_2	SO_2	NO_3 PO_4	C_2H_4	H_2O			MJ		

Impact categories are: GW = Global Warming, Ac = Acidification, Eu = Eutrophication, PO = Photochemical Oxidant, AE = Aquatic Ecotoxicity, TE = Terrestrial Ecotoxicity, HT = Human Toxicity, EU = Energy Use, BR = Biotic Resource Use, OD = Ozone Depletion. Category indicators are: CO_2 = Carbon Dioxide, SO_2 = Sulphur Dioxide, PO_4 = Phosphate, NO_3 = Nitrate, O_2 = Oxygen, 1,4 DCB = 1,4 Dichlorobenzene, H_2O = Water, MJ = Mega Joules, Sb = Antimony, NPP = Net Primary Productivity, CFC = Chlorofluorocarbon

have yet to be developed. A logical starting point, then, is to determine which impacts might be meaningfully quantified using the LCA framework, and which cannot. This will largely depend on two factors: the quality of available information regarding the impact in question; and the ability to link the impact to a functional unit in a realistic manner. In many cases, such as the quantification of benthic impacts, the development and application of an appropriate impact category appears promising but may be constrained by a lack of scientific consensus, data, and accepted models. In other cases, as when the impacts in question arise from a complex interplay of several variables and cannot be reduced to direct causal relationships, it may simply not be possible to defensibly link the impacts to a functional unit. Examples include changes to biodiversity, such as when seabird colonies exhibit declines attributable only in part to competition with fisheries for food resources, or declines in wild salmon stocks impacted by a variety of factors including sea lice emissions from net-cage aquaculture operations. Moreover, while a variety of local impacts may, indeed, prove relatively easy to quantify in relation to a functional unit, the value of incorporating these aspects into an LCA must be assessed with respect to the goals of the research. In all situations where quantitative analysis of specific impacts proves unfeasible, the value of qualitative analyses should be considered.

The following discussion is intended to review various approaches that seafood and other LCA researchers have used to attempt to quantify a range of non-traditional life cycle impacts, and to highlight potential areas for future impact category development efforts for seafood LCAs. Rigorous evaluations of the technical merits of specific approaches have not been undertaken.

2 Methodological Innovations

A review of published LCA research in fisheries and aquaculture indicates that modest efforts have been made to expand the suite of impact categories beyond those commonly employed in other sectors. In addition, several authors have discussed the desirability of new impact categories or described possible additional categories.

Ziegler et al. (2003) pioneered the use of seafloor effects due to bottom trawling as an impact category in a Life Cycle Assessment of frozen cod fillets. Seafloor effects were calculated by quantifying the area of seafloor swept by trawls per functional unit landed. Although this initial attempt to quantify benthic impacts was somewhat limited in that it only distinguished two seafloor types (oxic and anoxic) and did not differentiate between the associated benthic communities and their resilience to disturbance, considerable effort has since been expended to incorporate these complexities into the impact assessment method (Nilsson and Ziegler 2006).

In a more recent study, a Geographical Information System (GIS) was used to analyze the spatial distribution and biological impact of fishing effort in relation to seafloor habitat types. An index of gear impact per hour trawled was calculated from vessel speed as well as otter board and trawl width for the demersal trawl types in use. This index was multiplied by the fishing effort for each trawl haul to arrive at the total seafloor area swept during each fishing event. Fishing effort intensity was then modeled by dividing the area into squares in which the number of times swept was calculated, based on the gear set position and seafloor impact calculated as above. Next, this dataset was overlaid with the habitat map showing the fishing intensity in the individual habitats. It was found that large areas were not used for fishing during the study period (three years) and that the habitats were targeted to highly different degrees. The biological impact of the fishing intensities was assessed using a British database detailing marine habitat sensitivity and recoverability from fishing disturbance. Results which could be used as a seafloor impact indicator include the percentage of each habitat considered not to recover between fishing events (i.e. being in a permanently altered condition (Nilsson and Ziegler 2006).

Ziegler et al. (2003) also quantified by-catch of non-target and juvenile target species based on landing and discard data for the cod fishery in the Baltic Sea. A qualitative assessment of the sustainability of the resource use in relation to cod stock status was included. Given widespread concern regarding the ecosystem effects of by-catch in a diversity of fishing systems (Harrington et al. 2005, Hall and Mainprize 2005, Catchpole et al. 2005, Read et al. 2006), this category certainly merits further elaboration. It also provides a means of partially accounting for biotic resource use.

While many studies employ 1,4-dichlorobenzene as the category indicator for aquatic ecotoxicity, Ziegler et al. (2003) reported ecotoxicity in terms of volume of water polluted by the copper-based anti-fouling paints used on the hulls of fishing vessels. Thrane (2004a) and Hospido and Tyedmers (2005) similarly quantified anti-fouling paint emissions in LCA studies of Danish flatfish and Spanish tuna fisheries. These studies indicated non-trivial ecotoxicity impacts, suggesting the importance of modeling this emission in future LCA research of seafood production.

Papatryphon et al. (2004) used appropriation of Net Primary Production (NPP), which is the net flux of carbon from the atmosphere into green plants per unit time, as a proxy for biotic resource use impacts in an investigation of the life cycle impacts of salmonid feed production. For simplicity, analysis was restricted to direct NPP use, and did not include long-term effects on ecosystem NPP levels. In order to quantify global life cycle NPP use, a single category combining terrestrial and marine NPP was employed. For crop-based feed ingredients, NPP use was calculated according to the carbon content of the harvested and utilized crop components. NPP use for fisheries-derived ingredients was calculated based on the carbon content and trophic level of the species harvested.

The use of NPP to characterize biotic resource use represents one of the most interesting impact category developments to date. Appropriation of Net Primary Productivity provides a convenient and logical measure of biotic resource use. According to basic thermodynamic principles, energy can neither be created nor destroyed; all forms of energy are inter-convertible; but every transformation of energy results in increased entropy. These principles are evident in biological systems, almost all of which are driven by solar energy.

Photosynthetic plants and algae convert solar energy into heat (dissipated) and chemical (stored) energy in the form of carbon-based organic molecules. In turn, as these carbon complexes flow through trophic webs, the amount available to successive consumers decreases. Generally, only 10% of chemical energy is transferred between trophic levels in aquatic ecosystems (Pauly and Christensen 1995). This implies that food webs are effectively carbon-based energy pyramids, with high biomass of (numerous) primary producers at the bottom, and few, high-level consumers at the top. In this light, trophic dynamics can be interpreted as a competition for carbon resources, which represent the transferable products of primary production.

Human appropriation of global terrestrial NPP has been estimated at close to 40% (Vitousek et al. 1986), and an estimate of 8% has been advanced for the appropriation of marine NPP by fisheries (Pauly and Christensen 1995). The finite nature of primary production implies that optimal resource use in human food production systems must be informed by considerations of how best to allocate these resources between competing interests. This must include not only human needs, but the implications of such appropriation for biodiversity and ecosystem stability.

Measuring the appropriation of Net Primary Productivity provides opportunities to address allocative efficiency in fisheries and aquaculture on several counts. This metric can be used to incorporate the biophysical costs of by-catch in fisheries. It can also be used to elucidate the ecological demands of alternative aquafeed formulations, the culture of species occupying different trophic levels and, more generally, as a yardstick for comparing the relative ecological efficiency of different seafood production technologies. Such a measure is analogous to Ecological Footprint Analysis, which has been used to quantify the area of productive ecosystem support appropriated by various forms of aquaculture (Folke 1988, Larsson et al. 1994, Berg et al. 1996, Folke et al. 1998, Tyedmers 2000). One challenge in the use of this indicator is the conflict inherent in treating primary production as a limited resource while simultaneously pegging eutrophication as an environmental impact. In both cases, considerations should be made for local carrying capacity, and the status of the organisms in question in relation to baseline conditions.

Since traditional LCA research focuses on industrial production systems that consume primarily abiotic resources, the underdevelopment of appropriate measures for biotic resource use is somewhat understandable. This certainly does not hold for food production systems, where biotic resources are of central concern. Despite this, the majority of LCA studies of food products have failed to include any measure of biotic resource use – in effect, externalizing biotic resources by treating them as free system inputs.

In a LCA of Danish fish products, Thrane (2004b) reported an in-depth investigation of abiotic energy consumption in Danish fisheries. His analysis considered fuel consumption at the fisheries stage both as a function of fish species and fishing gear type. Fisheries for demersal fish and shellfish were found to be much more energy intensive than fisheries for pelagic fish for human consumption, mussels, and those fish destined for reduction to meal and oil. It was also concluded that significant reductions in fuel use could be achieved by using seine or gillnet gear instead of trawls in the Danish fishery. This is consistent with Tyedmers (2004), who reported that fuel use in fisheries can range from 20–3400 liters/tonne of fish landed. Hospido and Tyedmers (2005) similarly found that fuel inputs dominated environmental impacts in Spanish tuna fisheries, and concluded that efforts to rebuild stocks could result in improvements in the environmental performance of the fishery by decreasing distances traveled to catch and transport fish. In general, research of seafood production systems indicates that fuel use in the fisheries stage often contributes a disproportionate share of impacts, although the actual proportion can vary widely depending on the fuel-intensity of the fishery (Thrane 2006, Hospido and Tyedmers 2005, Tyedmers 2004, Ziegler et al. 2003). LCA research in aquaculture also suggests the importance of energy consumption to overall environmental impacts (Mungkung and Clift 2006, Mungkung 2005, Papatryphon et al. 2003, 2004). It would appear, then, that energy use should perhaps be treated as a stand-alone indicator of environmental performance in LCA research in the seafood sector – as, for example, was reported in Papatryphon et al. (2004).

Thrane (2004a) made qualitative evaluations of seabed impacts, land use, waste, use of non-renewable abiotic resources, use of groundwater, exploitation of fish, discards and by-catch, occupational health and safety, noise and accidents, and animal welfare. In a LCA of salmonid feeds, Papatryphon et al. (2004) expressed concern regarding the current lack of relevant impact categories, characterization factors and emission factors specific to the aquatic environment. Eutrophication effects, aquatic toxicity, Net Primary Productivity, damage to the benthos and biodiversity, the effects of solid waste, the use of chemicals, genetic impacts, and disease transmission were identified as areas requiring further research for impact category development. Mattsson and Ziegler (2004) also discussed the state of knowledge and research needs in regards to using seafloor effects, mortality of target and non-target species, working environment, anti-fouling agents, and fish welfare as impact categories for seafood LCAs. While these impacts are, indeed, highly relevant, convincing methods for relating them to a functional unit must be developed before they can be incorporated in standard LCA research.

3 Lessons from Agriculture

Although the creation of new impact categories for LCA has, in general, been limited, innovations and theory generated within the context of agri-food LCAs might usefully inform impact category developments for seafood. For example, Cowell and Clift (2000) discuss a methodology for assessing soil quantity and quality in Life Cycle Assessments of agricultural production systems. Relevant quantifiable factors describing changes to soil resulting from agriculture are identified. These include soil mass, nutrients, weeds and weed seeds, pathogens, nutrients, salts, pH, organic matter, and soil texture and structure. Given that water provides a comparable role as growth medium in fisheries and aquaculture, a similar suite of parameters could be developed to evaluate impacts to water from seafood production systems. Owens (2002) proposes a suite of detailed category indica-

tors for assessing water quantity and quality in Life Cycle Assessments that could provide a foundation for further refinement of a Water Resources impact category in seafood LCAs. Eutrophication, Dissolved Oxygen Demand, Thermal Effects, Pathogenic Micro-organisms, Colour and Turbidity, Suspended Solids, Toxic Hazard, and Effluent Toxicity are identified as potential indicators.

The Land Use category sometimes employed in agricultural LCAs might also have a potential parallel in fisheries and aquaculture. Just as the environmental costs of land use include the loss of habitat and potential associated loss of biodiversity, use of the marine environment for harvesting or producing seafood generates similar impacts. Brentrup et al. (2002) discusses the Hemeroby concept, which is a measure of human influence on ecosystems, as a possible quantifiable indicator for land use in agriculture. Hemeroby evaluates different types of land use according to intensity. Characterization factors can then be used to calculate landscape degradation, with reference to the natural state of the biogeographic region, due to specific land uses (Brentrup et al. 2002). In the context of seafood production, such characterization factors might include benthic macro-faunal diversity, zooplankton abundance and diversity, or sediment and water chemistry. However, it would first be necessary to define clear causal links between the described impact and the product system – a challenge compounded by the necessity of both extensive on-site sampling and considerable baseline information.

In a LCA study comparing organic, extensive and intensive grassland farming in southern Germany, Haas et al. 2001 employed several unconventional impact categories, including soil function, water quality, biodiversity, landscape image and animal husbandry. All of these are either directly applicable or have a potential parallel as impact categories for the marine environment should appropriate methodologies be developed.

4 Potential Socio-economic Impact Category Development

In addition to recent biophysical and ecological impact category developments, there is considerable potential for the creation of socio-economic impact categories (Dreyer et al. 2006) – the addition of which would much enhance the value of LCA in assessing and improving the sustainability of seafood production. However, despite the fact that SETAC standards recommend inclusion of 'social welfare' as an impact category in all detailed LCAs (SETAC, 1993), only a limited number of published studies have attempted to quantify the socioeconomic impacts (SEIs) associated with specific products/processes.

The integration of SEIs into the LCA framework is considerably less advanced for several reasons. The first is related to limitations inherent to the LCA framework, which was originally developed by industrial engineers to measure and mitigate impacts linked to specific flows of raw materials and energy. While these relationships are generally easy to establish, causal links describing relationships between a process and its socioeconomic impacts are more difficult to define. To date, a set of metrics suitable to describing such links for SEIs has not been established, nor is there a shared understanding as to how this is best achieved. O'Brien et al. (1996) advance a general framework for integrating Social and Environmental Life Cycle Assessment (SELCA) that might usefully inform the incorporation of social criteria in future impact category developments. In addition, a SETAC-Europe Working Group review of best practices for defining impact categories and category indicators provides further guidance (Udo de Haes et al. 1999). More recently, Dreyer et al. (2006) proposed a framework for Social Life Cycle Assessment (SELCA) based on a combined top-down and bottom approach that addresses the conduct of companies in relation to universal consensus documents such as the *Universal Declaration of Human Rights*, the *International Labour Organization's Conventions and Recommendations*, and the *Tripartite Declaration of Principles Concerning Multinational Enterprises and Social Policy*.

The successful integration of socioeconomic indicators into the LCA framework requires that impacts be measured using a metric that is additive along the value chain, or that they can be described in a way that facilitates comparisons between process steps. In others words, socio-economic impacts might be described quantitatively, in relation to the functional unit, or qualitatively – both of which can meaningfully inform inter and intra-framework comparisons.

Examples of SEI categories or indicators identified in several existing studies include: occupational health & safety, noise and accidents, and illegal workers and age of workers. However, these indicators and/or categories vary significantly in size, scope and metrics. A more useful and relevant way of classifying indicators might be to place them in categories of a similar size and scope. These impact categories could include: (1) Working conditions (including occupational health & safety, accidents), (2) Workforce (including illegal workers and age of workers), and (3) Business/company practice/influence (including corporate social responsibility).

Sabatella and Franquesa (2004) consider the use of SEIs in fisheries, but outside the context of an LCA framework. Similarly, Ellingsen (in Mattsson and Ziegler 2004) discusses the potential use of LCA to inform improvements to working environment in fisheries and aquaculture by reducing risk of fatalities. A recent agricultural LCA (Heller, 2002) featured the development of indicators to describe the sustainability of the U.S. food system (excluding seafood products), with different indicators developed for the various stages in the value chain. Some key socioeconomic indicators used in this study were 'percentage of illegal farm workers', 'increasing age of farm operators' and 'declining entry of young farmers'.

There are a variety of other socioeconomic sub-categories that can be used to describe impacts, but most topical LCA literature has focused on only one particular sub-category (e.g. Weidema, 2002), as opposed to developing a comprehensive set of SEIs. Only one existing LCA study of fisheries considers some integration of socioeconomic aspects (Thrane 2004a) by qualitatively describing Occupational Health & Safety and Noise and Accidents (including odor and visual aspects) as potential impact categories.

An exhaustive review of potential indicators of social welfare that might be used within the LCA framework is beyond the scope of this paper. However, attention to certain

broad social objectives relevant to food production can illuminate potential foci for impact category development. For example, if one accepts the premise that social welfare is positively influenced by a net gain in food security, then relative contributions to food security could serve as a quantifiable means of comparing the social costs and benefits of various food production systems. Using edible Energy Return On Investment (EROI) (Mitchell and Cleveland 1993) as a characterization factor, the LCA researcher could report net caloric gains or losses in relation to both fossil energy inputs and the appropriation of Net Primary Productivity. Refining this indicator to reflect net gains or deficits in food security within and between geo-political regions would also be important to evaluating how fisheries and aquaculture production systems contribute to the equitable distribution of resources. Moreover, such a measure would effectively bridge biophysical and socio-economic considerations, making visible the dynamic interplay of social preferences, human needs, and the supporting context provided by ecosystem goods and services.

Contributions to job security could similarly be used as a social welfare impact category. Where several production systems exist for the same product (for example, different fisheries targeting the same species), costs and benefits could be characterized according to the number of jobs created, human work-hours, reported accidents, and salary levels. This would also provide a quantitative frame of reference for comparing fisheries and aquaculture production systems generating similar products.

5 Conclusions

An assessment of the state-of-knowledge of Life Cycle Assessment, as it is applied to fisheries and aquaculture production systems, indicates significant potential for the use of this tool in promoting environmental and social improvements in these industries. As an analytic tool, LCA generates quantitative, biophysical data that can be used as a basis for making specific product/process improvements. More generally, it focuses attention on core sustainability issues related to eco-efficiency and broad-scale environmental impacts that are often overlooked in public discourse concerning the environmental interactions of fisheries and aquaculture production. In addition, as each reported Life Cycle Assessment draws from and contributes to a rapidly expanding information base and methodology, the potential for meaningfully informing biophysically-based management decisions increases.

However, the limited scope of existing impact categories significantly impairs the value of Life Cycle Assessment as a management tool. Further impact category development is therefore desirable to facilitate more comprehensive evaluations of the myriad environmental and social costs of seafood production. Methodological innovations from LCAs in the agri-food sector and the pioneering efforts of seafood LCA researchers provide sound guidance for future efforts, although the nature of this tool (which requires the establishment of direct, causal relationships between impacts and the product system that are measurable across process stages) limits the ability to quantify certain impacts within the LCA framework.

6 Recommendations and Perspectives

While the quantity of published LCA research for seafood production systems is clearly increasing, the influence this research will have on the ground remains to be seen. In part, this will depend on the ability of LCA researchers to advance methodological innovations that enable consideration of a broader range of impacts specific to seafood production. It will also depend on the ability of researchers to communicate with a larger audience than the current LCA community. Reporting mechanisms that transform Life Cycle Impact Assessment data into palatable and easily understood formats, and present these in relation to definable goals are required if the insights derived from LCA research in this sector are to be successfully applied to improve policy/management prescriptions. Researchers should make efforts to contextualize their results in terms of regional, national, or international standards or targets relevant to specific impact categories, and should carefully select impact categories with respect to the goals of the research project.

Acknowledgements. This work has been generously supported by the Lenfest Oceans Program of the Pew Charitable Trusts.

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Received: July 12th, 2006
Accepted: September 26th, 2006
OnlineFirst: September 26th, 2006